Winter emissions of carbon dioxide, methane and nitrous oxide from a minerotrophic fen under nature conservation management in north-east Germany

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SUMMARY

Drained peatlands are known to be important sources of carbon dioxide (CO$_2$) and nitrous oxide (N$_2$O). While CO$_2$ emissions occur mainly during the growing season, large N$_2$O emissions may occur during the non-growing season as well. Peatland re-wetting may be an effective measure to prevent those emissions. However, recent research shows that re-wetted peatlands may release large amounts of methane (CH$_4$) during the years immediately after re-wetting whereas abandonment of intensive grassland on drained peat soils possibly leads to low nutrient supply and thus to small greenhouse gas (GHG) emissions. Here we examine the role of extensification practices (such as abandonment of mineral fertilisation, reduced cutting frequency and a cattle-free winter period) on GHG emissions from a temperate peatland during winter. From November 2009 to March 2010 GHG measurements were made on a minerotrophic fen five years after intensive grassland use was abandoned. During the measurement period CO$_2$ and N$_2$O emissions amounted to 4.4 t ha$^{-1}$ and 2.6 t ha$^{-1}$ CO$_2$-equivalent, whilst CH$_4$ emissions were negligible. Altogether the site emitted 7 t ha$^{-1}$ CO$_2$-equivalent, of which 37 % was N$_2$O, even though the winter 2009/2010 was extraordinarily cold. Thus, extensification of grassland use alone may not be sufficient to reduce GHG emissions from temperate peatlands.

KEY WORDS: peatland; greenhouse gas emissions; extensive grassland

INTRODUCTION

The greenhouse gases (GHG) carbon dioxide (CO$_2$), methane (CH$_4$) and nitrous oxide (N$_2$O) contribute 63 %, 18 % and 6 % respectively to the global anthropogenic radiative forcing (Forster et al. 2007). In addition, N$_2$O plays an important role in the stratospheric ozone chemistry because it is an important source for nitric oxide (NO) and nitrogen dioxide (NO$_2$) in the upper atmosphere (Crutzen 1970). Owing to its relative stability it will remain the major ozone-depleting substance throughout the 21st century (Ravishankara et al. 2009).

Peatlands are known to be an important element of the global GHG cycles (Frolking et al. 2006). Although covering only 3 % of the world’s land-surface area (Lappalainen 1996), they store 20 % to 30 % of the soils’ C and N reserves (Martikainen et al. 1993). Natural peatlands are regarded as long-term sinks of C, converting atmospheric CO$_2$ to growing peat whilst emitting significant amounts of CH$_4$. Their net climatic impact is estimated to be a slight warming (if the effects of CH$_4$ emissions exceed those of carbon sequestration) or a slight cooling (if sequestration exceeds CH$_4$ emission), and depend on the time since peatland formation (Frolking et al. 2006). In contrast, drained peatlands act as a source of carbon and nitrogen, emitting CO$_2$ and N$_2$O from decomposing peat. Their net climate impact is a strong warming, with CO$_2$ effluxes up to 50 t ha$^{-1}$ a$^{-1}$ and N$_2$O effluxes up to 60 kg ha$^{-1}$ a$^{-1}$ (Couwenberg et al. 2011). Therefore, recently, re-wetting has been used as a measure to restore the peatlands’ function as a C and N sink, whilst also having high value in terms of nature conservation. However, associated studies indicate that flooding of eutrophic drained fens may cause strong CH$_4$ emission peaks, possibly counteracting the reduction of CO$_2$ and N$_2$O emissions (Höper et al. 2008, Wilson et al. 2009, Glatzel et al. 2011). It is suggested that these enhanced emissions may be caused by anaerobic consumption of organic litter formed by plants that died back after flooding rather than by anaerobic consumption of peat (Hahn-Schöffel et al. 2010).

Nevertheless, drained eutrophic fens under agricultural use have a great potential to emit large fluxes of GHGs. Their potential to emit N$_2$O is significantly greater than that of virgin fens or drained but nutrient-poor peatlands (Regina et al. 1996). N$_2$O emissions are usually driven by a combination of several factors. Background emissions are controlled by long-term site-specific conditions such as nutrient status (e.g. C/N quotient,
Maljanen et al. 2009), hydrological characteristics (Freeman et al. 1993, Martikainen et al. 1993), vegetation type (Glatzel et al. 2008) and soil temperature (Röver et al. 1998). Event-based emission peaks are induced by short-term changes of site-specific conditions such as freeze-thaw cycles (Teepe et al. 2000), fertiliser application (Ruser et al. 2001) and heavy rain. Hence, the annual release of N₂O can be very erratic, with a very large temporal and spatial variability (Flessa et al. 1995).

Despite this strong variability, non-growing-season effluxes may contribute 40–80 % of the annual emission of nitrous oxide on boreal minerotrophic peatlands (e.g. Alm et al. 1999, Regina et al. 2004, Maljanen et al. 2009) and temperate mineral soils (e.g. Flessa et al. 1995, Röver et al. 1998, Teepe et al. 2000). In contrast, studies of winter nitrous oxide emissions from temperate peatlands are scarce. Those existing either find a winter contribution of about 50 % (Beek et al. 2010) or N₂O effluxes too close to the detection limit to be further analysed (Hendriks et al. 2007).

In contrast to nitrous oxide, there are many reports of winter carbon dioxide and methane emissions from temperate and boreal wetlands. These emissions are significantly smaller outside the growing season (Dise 1992, Melloh & Crill 1996, Alm et al. 1999, Panikov & Dedysy 2000, Hao et al. 2006, Hendriks et al. 2007, Beek et al. 2010), contributing 10–40 % (CO₂) or about 10 % (CH₄) depending on site-specific climatic conditions during the winter.

In Mecklenburg-Western-Pomerania (northeastern Germany) an area of 245,152 ha is covered by peatlands (Zauf et al. 2010), which account for 10.6 % of the federal state’s land surface. By the early 1990s, 99 % of these peatlands were drained for agricultural use (Gelbrecht et al. 2001). It is now an integral part of Mecklenburg-Western-Pomerania’s environmental policy to protect and restore peatlands. An important goal is the reduction of GHG emission and the calculation of its impact to be further analysed (Hendriks et al. 2007).

In the work reported here we measured the effects of extensive agricultural use on GHG effluxes from a drained eutrophic fen.

**METHODS**

**Site description**

The study site (108 ha) is part of the Biosphere Reserve “Schaalsee”, in a small river valley (“Neuenkirchener Niederung”) near Lake Schaalsee, 60 km east of Hamburg (Mecklenburg-Western Pomerania, northern Germany; 53°36’ N, 10°59’ E, see Figure 1). The regional climate is temperate with a maritime influence. Climatic data were derived from a 1 × 1 km grid provided by the German Weather Service. The data were extrapolated using data from nearby stations and a digital elevation model, and have been shown to correlate well with measured values (Müller-Westermeier 1995). Mean annual air temperature is 9.0 °C. The long-term mean temperature in January is 0.2 °C, and the average snow cover period is 5.9 days. Annual precipitation is 711 mm with an annual climatic water balance of +134 mm (Hippeke, pers. comm.).

The fen is part of a river valley mire system that originated from a glacial tunnel valley. As its hydrology is dominated by groundwater flow, it is a percolation mire (Joosten & Succow 2001). The Bek, a dredged and channelled river, flows through the mire system. The surrounding area of the study site is characterised by agricultural land use on stagnosols originating from glacial till. These soils are typically decalcified down to 30–50 cm (Damm & Ratzke 2004) with pH 4–6 (Reuter 1962).

The study site was moderately drained during the 19th century. During the second half of the 20th century much deeper ditches were made (1.5–2.2 m), profoundly altering the water balance of the system. These ditches guide the main part of the outflow around the fen through a major ditch (L157), that formerly ran into the Bek (Figure 1).

From the 1970s until 2004 the study site was intensively used as grassland. Since then, the area has been managed under nature conservation guidelines, including abandonment of mineral fertilisation, reduced cutting frequency and a cattle-free winter period. Typical plant species of intensively used temperate grasslands are still dominant (such as *Alopecurus pratensis* L. and *Poa*...
Further species with large abundance or coverage are *Poa pratensis* L., *Taraxacum officinale* L., *Ranunculus repens* L. and *Holcus lanatus* L. indicating an ample nutrient supply from the soil (Ellenberg & Leuschner 2010). The plant community can be classified as *Molinio-Arrhenatheretea* Tx. 1937. The fen peat reaches more than 5 m deep, but the upper layer of peat (1 m) is strongly decomposed (H10–H8, von Post scale). At depths where decomposition is not too strong, peat originating from alder can be found at the western and at the eastern boundaries of the fen, whereas in the central parts the peat originates from reed and sedges (Table 1). Re-wetting of the mire system is planned but not yet implemented.

**Study design**

In October 2009, four GHG measurement plots were established along a transect crossing the “Neuenkirchener Niederung” from west to east. The plots were chosen to span the expected water conditions caused by the planned re-wetting of the mire (Figure 1). Plot 2 represents the wettest and Plot 4 the driest conditions, while Plots 1 and 3 will have intermediate water levels. In 2009, the area lying west of the Bek (Plot 1) was grazed by young male beef cattle. The area east of the Bek (Plots 2, 3, 4) was cut twice during the same year. We measured gas fluxes in removable chambers set on collars sunk permanently into the peat surface. We could not install these collars until the cattle had been removed from the plots, only three weeks before measurements began. The good fit of CO₂ effluxes versus soil temperature (see Figure 5) indicates, however, that any altered respiratory activity (due to root damage caused by installation of the collars) was negligible. Because of grazing, mowing and the protected status of the fen, no system of boardwalks could be laid; but the peat was dense and strongly humified, so ebullition events caused by physical disturbance when the chambers were approached were neither expected nor observed. The absence of stepwise linear increases in concentration, that would have been a sign of ebullition events (Chanton & Whiting 1995), supports this conclusion.

Each plot consisted of a triangle of collar locations (about 5 m apart). Since the nature of N₂O effluxes is erratic, an unfeasibly large number of collars would be needed to achieve statistically
Table 1. Characteristics of Plots 1 to 4: degree of decomposition (von Post scale); organic carbon (OC) proportion of dry mass [g/g] estimated from a 1m peat core; C/N quotient estimated from cores (n=5) of the uppermost 0.3 m.

<table>
<thead>
<tr>
<th>Plot</th>
<th>identified macrofossils</th>
<th>degree of decomposition</th>
<th>OC</th>
<th>C/N</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>alder</td>
<td>H8–H10</td>
<td>0.67</td>
<td>12.3</td>
</tr>
<tr>
<td>2</td>
<td>reed</td>
<td>H8–H10</td>
<td>0.68</td>
<td>13.3</td>
</tr>
<tr>
<td>3</td>
<td>reed/sedges</td>
<td>H9–H10</td>
<td>0.57</td>
<td>12.4</td>
</tr>
<tr>
<td>4</td>
<td>alder</td>
<td>H8–H10</td>
<td>0.41</td>
<td>11.8</td>
</tr>
</tbody>
</table>

significant differences of N₂O effluxes between treatments (Folorunso & Rolston 1984). Thus, our approach was a compromise between covering spatial variability and being able to make all the necessary measurements during the short daylight hours in winter. The circular PVC collars (30 cm outside diameter, 20 cm tall, 8 mm thick) were inserted in slots cut in the peat with a knife (depth between 5 and 10 cm). GHG effluxes were estimated from concentration measurements using the non-steady-state chamber method (Livingston & Hutchinson 1995). Sampling was carried out every two weeks from November 2009 until March 2010, with additional sampling during freeze-thaw events. Overall, the measurement period lasted 129 days.

For each sampling, opaque PVC chambers (diameter 30 cm, height 30 cm) were carefully placed on top of the collars. The collar-chamber edge was sealed with a grooved ring from the inside and taped from the outside. Snow within the collar was not removed as snow removal is known to alter GHG effluxes (Maljanen et al. 2003, 2009). Four gas samples were taken (one every 15 minutes) with evacuated gas flasks (100 ml) that were attached to the chambers with a short (≤ 5 cm) silicone rubber tube. The samples were analysed by gas chromatography (Perkin Elmer Auto System) within a week for concentration of CO₂, N₂O and CH₄ using an Electron Capture Detector (ECD) and a Flame Ionization Detector (FID). The precision of analysis was about 10 vpb for methane, 70 vpb for nitrous oxide and 10 vpm for carbon dioxide.

Furthermore, we measured soil and air temperature, depth of water table, soil water, and nitrate concentration during each sampling period. High-resolution meteorological data (air temperature, precipitation, relative humidity, air pressure) covering the measurement period were collected by a weather station in Zarrentin, 7 km south-east of the study site. Peat characteristics and a height/depth profile along the transect were recorded once.

Data analysis
Gas efflux rates were estimated from the chamber concentration data using a prototype version of the R package “flux” (Jurasinski & Koebsch 2011). We used it to obtain the best linear fit to any three points out of the four possible groups of three (abc, abd, acd, bcd). The parameters of the model with the best linear fit (greatest R²) were then used to obtain the change in concentration in the chamber headspace over the sampling time (dc/dt). When none of the models had R² ≥ 0.8 the resulting efflux was discarded. The gas effluxes were calculated according to Fick’s first law and the assumption that diffusion is the single process of gas accumulation in the chamber headspace. Thus, the efflux rate \( f \) (mg m⁻² h⁻¹ for CO₂, µg m⁻² h⁻¹ for CH₄ and N₂O) was calculated from the molar mass \( M \) (g mol⁻¹) of the gas, the air pressure \( p \) (Pa), the chamber volume \( V \) (m³), the gas constant \( R \) (m³ Pa K⁻¹ mol⁻¹), the chamber temperature \( T \) (K), the surface area \( A \) (m²) and the concentration change over the sampling time \( dc/dt \) (vpm h⁻¹ for CO₂, vpb h⁻¹ for CH₄ and N₂O) as follows:

\[
f = 10^3 \frac{M p V}{R A T} \frac{dc}{dt}
\]  

Plot-wise efflux was calculated as the mean efflux of the three chambers. Estimation of total efflux rates during the sampling period was made by integrating the area under the efflux curves. To calculate the global warming potential (in CO₂-equivalents) we used the 100 year time horizon given by Forster et al. (2007) with 25 CO₂-eq for CH₄ and 298 CO₂-eq for N₂O.

Differences of efflux and of environmental variables among the four plots were tested for significance using the pair-wise Wilcoxon rank test with Bonferroni adjustment of \( P \)-values, because the data within single plots were not normally distributed in all cases. Generalised linear models...
were constructed to explain the variability within plots, and mixed effect models were built to explain the variability between plots. The best model was found by step-wise deletion of non-significant parameters (Crawley 2005). All statistical analyses were performed with R 2.12.0 (R Core Development Team 2011).

RESULTS

Winter 2009/2010 environmental characteristics

The winter of 2009/2010 was the harshest for 30 years in Mecklenburg-Western Pomerania. The monthly mean temperatures of December, January and February fell below the long-term mean temperatures (1970–2000) by 1.6, 4.4 and 1.4 °C, respectively. The lowest air temperature (−17 °C) was measured around 26th January. The long-term average snow-cover period of 6 days per winter was exceeded by more than 60 days (Figure 2). The snow cover reached a maximum thickness of about 40 cm after heavy snowfall at the end of January and lasted until the melting period one month later. During this period, the snow cover isolated the soil from fluctuating air temperature. Therefore, soil temperature remained constantly around 0 °C and the eastern area of the fen (Plots 2, 3, 4) continued to be unfrozen during the snow-cover period. In contrast, a strong freeze-thaw cycle occurred just when the mire became snow free again. At that time, the air temperature varied greatly between day and night ranging from +4 °C to -10 °C (Figure 2).

The water table depths during the measurement period differed slightly among the four plots according to the expected water table conditions after a possible re-wetting (Figure 1, Figure 4). The depth of water table at Plot 2 was significantly smaller than at Plot 4 (according to Wilcoxon rank test at $P < 0.05$). Water table depths at Plots 1 and 3 did not differ significantly from each other or from those at Plots 2 and 4 during the measurement period. Furthermore, parts of the study area (Plot 2, similar to “inundation at mean runoff”, Figure 1) were inundated by meltwater for two weeks from the beginning to the middle of March 2010. The soil temperature was similar in all four plots throughout the investigation period.

Gas effluxes

Greenhouse gas effluxes occurred during the whole measurement period even when the ground was snow-covered (Figure 3). The greatest effluxes of CO$_2$ and N$_2$O were recorded in November and March during periods with relatively high temperatures and without snow cover. At plot scale, GHG effluxes usually approximated a normal distribution except CO$_2$ effluxes at Plot 2 (Table 2).
N\textsubscript{2}O effluxes were similar at Plots 1, 2 and 3 (-380 to 420 µg m\textsuperscript{-2} h\textsuperscript{-1}) but significantly greater at Plot 4 (310 to 1900 µg m\textsuperscript{-2} h\textsuperscript{-1}) (Figure 4). On 10\textsuperscript{th} March an N\textsubscript{2}O efflux peak was observed at Plot 4 reaching 1400 µg m\textsuperscript{-2} h\textsuperscript{-1}. It followed a week-long freeze-thaw cycle that reached minimum temperatures of -10 °C at night and maximum temperatures of +4 °C during daytime.

CO\textsubscript{2} effluxes were similar at all four plots, ranging from a small uptake of -160 mg m\textsuperscript{-2} h\textsuperscript{-1} to 650 mg m\textsuperscript{-2} h\textsuperscript{-1}. CH\textsubscript{4} emissions were near zero throughout the measurement period except for

Figure 3. Gas effluxes and depth of water table (bottom row) at Plots 2 and 4 during the measurement period. These data span the hydrological range of the transect (Plots 1 and 3 have intermediate water level conditions, see Figure 4). For gas measurements, dots indicate single measurements and black lines indicate curves of mean effluxes. The zero line (grey) of the water table graphs marks the soil surface. Note the differences in scales.
Plot 2 at the end of November and for Plots 1 and 2 at the end of March after the snow-cover period. CH₄ effluxes ranged from -90 to 490 µg m⁻² h⁻¹ but did not differ significantly among plots (Figure 4).

CO₂, CH₄ and N₂O effluxes contributed 63 %, 0.2 % and 37 %, respectively, of the accumulated greenhouse gas emissions of the study site (Table 3). N₂O played the most important role.

Table 2. Characteristics at four plots of gas effluxes during the 129-day measurement period November 2009 to March 2010. Mean and median effluxes and coefficients of variation (CV) are given. Non-normally distributed effluxes (according to Shapiro-Wilk test at $P < 0.05$) in italics.

<table>
<thead>
<tr>
<th>Plot</th>
<th>CO₂ (mg m⁻² h⁻¹)</th>
<th>Median</th>
<th>CV</th>
<th>CH₄ (µg m⁻² h⁻¹)</th>
<th>Median</th>
<th>CV</th>
<th>N₂O (µg m⁻² h⁻¹)</th>
<th>Median</th>
<th>CV</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>180</td>
<td>188</td>
<td>0.71</td>
<td>37</td>
<td>0</td>
<td>1.60</td>
<td>31</td>
<td>-28</td>
<td>3.92</td>
</tr>
<tr>
<td>2</td>
<td>227</td>
<td>150</td>
<td>0.81</td>
<td>118</td>
<td>42</td>
<td>1.49</td>
<td>24</td>
<td>47</td>
<td>5.45</td>
</tr>
<tr>
<td>3</td>
<td>132</td>
<td>124</td>
<td>0.99</td>
<td>11</td>
<td>11</td>
<td>1.37</td>
<td>26</td>
<td>10</td>
<td>9.58</td>
</tr>
<tr>
<td>4</td>
<td>198</td>
<td>203</td>
<td>0.66</td>
<td>-6</td>
<td>-14</td>
<td>-4.21</td>
<td>1013</td>
<td>896</td>
<td>0.55</td>
</tr>
</tbody>
</table>

Figure 4. Plot-wise distribution of greenhouse gas effluxes and water table levels (relative to ground surface) during the measurement period. Thick line: median; box extent: interquartile range; whisker extent: marks the data lying within 1.5 times interquartile range. Different lower case letters indicate significant differences between the plots at $P < 0.05$, according to pair-wise Wilcoxon-rank-test with Bonferroni $P$-value adjustment. Note the small number of samples ($n < 12$) for each plot.
Table 3: Total emissions from four plots of carbon dioxide, methane and nitrous oxide (t ha\(^{-1}\) CO\(_2\)-equivalents), contributions of CO\(_2\), CH\(_4\), N\(_2\)O (in %, *italics*) and mean depth of water table during the measurement period.

<table>
<thead>
<tr>
<th></th>
<th>all plots</th>
<th>Plot 1</th>
<th>Plot 2</th>
<th>Plot 3</th>
<th>Plot 4</th>
</tr>
</thead>
<tbody>
<tr>
<td>CO(_2)</td>
<td>4.4</td>
<td>3.9</td>
<td>5.7</td>
<td>2.9</td>
<td>5.2</td>
</tr>
<tr>
<td></td>
<td>63</td>
<td>85</td>
<td>102</td>
<td>84</td>
<td>36</td>
</tr>
<tr>
<td>CH(_4)</td>
<td>0.02</td>
<td>0.02</td>
<td>0.04</td>
<td>0.01</td>
<td>0.00</td>
</tr>
<tr>
<td></td>
<td>4.7</td>
<td>2.6</td>
<td>0.7</td>
<td>-0.2</td>
<td>0.6</td>
</tr>
<tr>
<td>N(_2)O</td>
<td>2.6</td>
<td>0.7</td>
<td>-0.2</td>
<td>0.6</td>
<td>9.2</td>
</tr>
<tr>
<td></td>
<td>37</td>
<td>14</td>
<td>-3</td>
<td>16</td>
<td>64</td>
</tr>
<tr>
<td>Mean depth of water table (cm)</td>
<td>-15</td>
<td>-12</td>
<td>-3</td>
<td>-18</td>
<td>-28</td>
</tr>
</tbody>
</table>

(64 %) at Plot 4, whereas at Plot 2 N\(_2\)O was taken up during the measurement period. Plot 4 had the greatest release of GHG, emitting more than 14 t ha\(^{-1}\) CO\(_2\)-equivalents, whilst Plot 3 had the smallest (3.5 t ha\(^{-1}\)). On average, the study site emitted 7.0 t ha\(^{-1}\) CO\(_2\)-equivalents during the winter of 2009/2010.

**Environmental controls**

Carbon dioxide and methane effluxes did not differ among the plots (Figure 4). Therefore, generalised linear regression models were built by using environmental data covering the whole measurement period (Figure 5). Carbon dioxide efflux \(R_{\text{CO}_2}\) (mg m\(^{-2}\) h\(^{-1}\)), in relation to soil temperature \(T_{\text{soil}}\), can be described best (among the models tested) by:

\[
R_{\text{CO}_2} = R10^{ \left( E0 \left( T_{\text{soil}} - 227.13 \right) \right) \left( \left( 283.15 - 227.13 \right) \left( T_{\text{soil}} - 227.13 \right) - 1 \right) }
\]  
(\(R^2 = 0.55, P < 0.001\), following the suggestion of Lloyd & Taylor (1994), with \(R10 = 429.7\) mg m\(^{-2}\) h\(^{-1}\)) being the reference efflux at 10 °C and \(E0 = 392.1\) K.

Methane effluxes at the study site during the winter season can be best described by the following regression equation:

\[
y = 288.6 - 91.3 \cdot \ln(bx) \]  
\( (R^2 = 0.38, P < 0.001)\) in which \(y\) is the CH\(_4\) efflux (µg m\(^{-2}\) h\(^{-1}\)), \(x\) is the depth of water table (cm), and \(b = 1\) cm\(^{-1}\).

In contrast, nitrous oxide effluxes do differ between the plots (Figure 4). Therefore, the model was built using parameters that discriminate the plots in space. The best model found is:

\[
z = 129.4 + 48.6 \cdot bx - 74.3 \cdot bx : w \]  
Figure 5. Relations of CO\(_2\)- (\(R^2=0.55, P < 0.001\)) and CH\(_4\)-fluxes (\(R^2=0.38, P < 0.001\)) to the relevant ecosystem controls. A circle represents the mean of plot-wise efflux measurements \((n=3)\) with circle diameter being proportional to a third variable, namely depth of water table in (A) and air temperature in (B).
this reason, the N\textsubscript{2}O emissions measured in this
winter, the N\textsubscript{2}O emissions of the Neukenkirchen
Niederung are still as large as reported for
intensively used grassland on organic soils (Velthof
authors show that fertilisation is not necessarily
needed on peatlands to produce N\textsubscript{2}O effluxes as
much as 10,000 \(\mu g m^{-2} h^{-1}\) (Maljanen et al. 2009)
or
56.4 kg ha\(^{-1}\) a\(^{-1}\) N\textsubscript{2}O-N (Flessa et al. 1998). Depth of
water table and organic carbon content may be
much more important drivers of N\textsubscript{2}O emissions in
our case (Equation 4). These were shown to be
important controls in several other studies (e.g.
Freeman et al. 1993, Martikainen et al. 1993,
Maljanen et al. 2009).

\textit{DISCUSSION}

The radiative forcing of GHG emissions from the
Neukenkirchen Niederung during the winter
2009/2010 was mainly caused by carbon dioxide
(63 \%) and nitrous oxide emissions (37 \%). Methane emissions played only a minor role.

Compared with other studies, the global warming
potential (GWP) of winter N\textsubscript{2}O emissions (2.6 t ha\(^{-1}\)
CO\textsubscript{2}-equivalents) was similar to the GWP of annual
N\textsubscript{2}O emissions reported from a drained fen in
Finland (2–4 t ha\(^{-1}\), Nykänen et al. 1995) and to the
GWP of winter N\textsubscript{2}O production of organic grassland
soils in the Netherlands (~2.8 t ha\(^{-1}\) a\(^{-1}\), derived from
Beek et al. 2010). Given the extraordinarily cold
winter of 2009/2010 with its long period of snow
cover, and that the greatest nitrous oxide emissions
occurred in late autumn (November) and early
spring (March), our results resemble an emission
pattern typical for a boreal climate as described by
Alm et al. (1999) rather than an emission pattern for
a temperate climate where the largest N\textsubscript{2}O emissions
may occur during the winter months of December to
February (Flessa et al 1995, Teepe et al. 2000). For
this reason, the N\textsubscript{2}O emissions measured in this
study seem likely to have been smaller than during a
climatically normal winter.

The snow cover during the winter of 2009/2010
prevented the soil from becoming deeply frozen. In
such cases, nitrous oxide effluxes during freeze-and-
thaw events are typically smaller than effluxes from
formerly deeply frozen soils (Maljanen et al. 2009).
In contrast, effluxes during the snow-cover period
are typically greater if the soil is not deeply frozen.
For this reason, the distribution of N\textsubscript{2}O effluxes in
this study may not be as skewed as reported by
others (Flessa et al. 1995). Hence, the timing of
snow cover development and frost during the
beginning of winter is an important factor that
controls N\textsubscript{2}O emissions in the non-growing season
(Maljanen et al. 2003). Nevertheless, an N\textsubscript{2}O efflux
peak of 1,400 \(\mu g m^{-2} h^{-1}\) at Plot 4 was detected after
one week of intensive freezing and thawing of
snow-free soil. Phillatie et al. (2010) reported a time
span of one week after freezing and thawing to be
the period of greatest N\textsubscript{2}O release during such a
freeze-thaw cycle.

Although the land use of the study site was
extensified five years before the measurements were
carried out and despite the extraordinarily cold
water, the N\textsubscript{2}O emissions of the Neukenkirchen
Niederung are still as large as reported for
intensively used grassland on organic soils (Velthof
authors show that fertilisation is not necessarily
needed on peatlands to produce N\textsubscript{2}O effluxes as
much as 10,000 \(\mu g m^{-2} h^{-1}\) (Maljanen et al. 2009)
or
56.4 kg ha\(^{-1}\) a\(^{-1}\) N\textsubscript{2}O-N (Flessa et al. 1998). Depth of
water table and organic carbon content may be
much more important drivers of N\textsubscript{2}O emissions in
our case (Equation 4). These were shown to be
important controls in several other studies (e.g.
Freeman et al. 1993, Martikainen et al. 1993,
Maljanen et al. 2009).

\textit{CO\textsubscript{2} efflux can be modelled with our data}
following Lloyd & Taylor (1994) and is thus driven
by soil temperature. Although soil temperature
varied only within an interval of 10 °C during the
measurement period, a clear dependence of CO\textsubscript{2}
efflux on soil temperature was observed
(Equation 2). Since grassland use has been constant
for decades and the study site is either used for hay
production or cattle grazing, C uptake of the soil
during the growing season is improbable. Therefore,
the CO\textsubscript{2} balance of the system can be described by
CO\textsubscript{2} emissions alone showing that the study site acts
as a source for CO\textsubscript{2}. The winter CO\textsubscript{2} emissions from
the study site amount to 4.4 t ha\(^{-1}\), which is closer to
the winter CO\textsubscript{2} emissions of a boreal minerotrophic
fen (2.5 t ha\(^{-1}\), derived from Alm et al. 1999) than to
the winter CO\textsubscript{2} emissions of an abandoned maritime
peat meadow (11.0 t ha\(^{-1}\), derived from Hendriks et
al. 2007), again indicating the impact of the
extraordinarily cold winter 2009/2010. In contrast,
the CO\textsubscript{2} balance from virgin fens is generally close
to zero or even negative (Frolking et al. 2006) as
their hydrology inhibits oxic decomposition, and
carbon is sequestered.

Winter methane effluxes were close to the
detection limit and contributed less than 0.2 \% (0.02 t ha\(^{-1}\) CO\textsubscript{2}-equivalents) to the total winter
emissions of the study site. Annual methane effluxes
from similar fen sites in Finland (~0.04 t ha\(^{-1}\),
derived from Nykänen et al. 1995) are in the same
order of magnitude. In contrast, annual methane
emissions from virgin fen sites are two orders of
magnitude greater (8–15 t ha\(^{-1}\) CO\textsubscript{2}-equivalents,
derived from Dise 1992, Nykänen et al. 1995,
Melloh & Crill 1996), but N\textsubscript{2}O emissions from such
virgin sites are typically close to the detection limit.

To get an idea about the magnitude of the emission
potential from the study site we assumed
that winter carbon dioxide emissions contribute
25 \% (23 \% according to Alm et al. 1999, 34 \%,
derived from Hendriks et al. 2007), winter methane
emissions contribute 10 \% (Dise 1992, Alm et al.
1999) and winter nitrous oxide emissions contribute
Therefore, the annual GHG emissions of the study site are estimated at 18 (CO₂), 0.2 (CH₄) and 5 (N₂O) t ha⁻¹ CO₂-equivalents, which would total about 23 t ha⁻¹, with the winter contributing 30 %. For this reason, our assumption of small net GHG emissions during winter must be rejected. This indicates that winter GHG emissions from temperate peatlands should be taken into account when comparing the GHG emission potentials of extensively used and re-wetted mires.

In addition, both young male beef cattle and dairy cattle emit methane. Given that one lactating cow (weight 650 kg, milk yield 6500 kg a⁻¹, CH₄ emission 135 kg a⁻¹, Jentsch et al. 2009) needs about 0.8 ha of extensively used grassland, the net annual methane emissions of the study site would increase to about 4 t ha⁻¹ CO₂-equivalents, and the annual GHG emissions to 27 t ha⁻¹. Therefore, extensification of land use without re-wetting might not reduce the GHG emissions of the Neuenkirchener Niederung. According to our findings, aiming to reduce the GWP of peatlands is not a suitable objective. Only a permanent and effective rise of the water table to a level close to the ground surface will lead to permanently reduced CO₂ and N₂O emissions.

However, re-wetting of drained fens may cause raised CH₄ emissions that possibly counteract the reduced CO₂ and N₂O emissions (Höper et al. 2008, Wilson et al. 2009, Glatzel et al. 2011). Field studies of GHGs from these dynamic and young ecosystems are few, and seldom cover long time spans. On the one hand, increased methane emissions after re-wetting might originate from dead inundated grassland plant such as Phalaris arundinacea L. (Hahn-Seböl et al. 2010). On the other hand, typical wetland plants such as Typha spp. or Phragmites australis L. are absent directly after re-wetting, but they are known to be important vectors for CH₄ emissions in virgin peatlands (Chanton et al. 1993, Van der Nat et al. 1998).

Thus, it can be assumed that these large CH₄ emissions prevail after re-wetting as long as the plant composition of the ecosystem is shifting towards a new equilibrium state reflecting altered hydrological conditions of the habitat. On a raised bog in north-east Germany, Bönsel & Sonneck (2011) showed that this shift in plant composition after re-wetting takes at least a decade. Therefore, we propose that future research on GHG emissions from peatlands should focus:

1) on study sites at different stages of re-wetting, in order to develop a time series model of CH₄ emission dynamics after re-wetting; and

2) on the estimation of CH₄ emission potentials of different vegetation types from drained peatlands, in order to better predict the possible CH₄ output of inundated grassland plant communities.

When these points are addressed, comparisons of GWPs of drained and re-wetted peatlands should be more accurate and reliable than comparisons of annual GHG budgets derived from single- or two-year measurement campaigns.

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